DETERMINATION OF WATER QUALITY IN LAKES BY BIOLOGICAL INDICES

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Some problems of evaluation of water quality by biological indices which can be applied in the practice of ecological monitoring on water bodies are considered in this report. Taking into account, that ecological monitoring is the most urgent for large lakes, situated in civilised (urbanised) and (or) agrarian landscapes the corresponding problems will be considered mainly in conformity with large deep lakes of temperate latitudes.

Numerous investigations on the problem of biological indication of water quality are being conducted in many countries and different practical methods have been reported in the literature. However, none of the proposed methods is without defects and is universally accepted.

Our aim is a general evaluation of some of the methods from the point of view of their possible application for monitoring on large water bodies. Before considering the methods we should like to call attention to the fact that ecological monitoring on water bodies of this size class cannot be sufficiently effective (using any method) unless the specific character of the structure and functioning of the ecosystem in question is taken into account.

In connection with this it is expedient to consider the following aspects.

1. Main elements of the specific limnological character of the large deep lakes of temperate latitudes.

2. Heterogeneity of the ecosystem of large lakes and conditions of water quality formation.

3. Mechanisms of ecosystem stability under varying external conditions.

4. Main restrictions of the existing methods of biological indication of water quality for their use on large lakes.

5. Integral biological indices of water quality and technical potentialities for ecological monitoring in heterogenic lake ecosystems.

1. In principle, the same physical and biological (ecological) laws are acting in large and small natural water bodies, but some of them take on an essential limnological significance only at a definite scale of a system ie size of a water body (Nikolaev, 1975). For example, the inertia of thermal processes in large water bodies is not only quantitatively manifested to a greater extent, but it is displayed in specific structures of the hydrobiological regime – in thermal and density stratification of the water in summer and in the formation of a thermal bar in spring and autumn. Both the phenomena may be also revealed on rare occasions in small water bodies, but they are, on one hand, only of short duration, and not substantial in limnological importance on the other hand.

The inertia of thermal processes also determines the seasonal heterolimny of large deep lakes. In the lakes Ladoga and On ega, in the spring – summer period, the warming of the water and the changes in the seasonal biological phases connected with it are delayed by 1-1.5 months in the deep water by comparison with the shallow zone. In L. Baikal this lag even is one of 2--2.5 months.

The phenomenon of upwelling is to some extent notable in small continental water-bodies but its essential limnological (and ecological) function is displayed only in large deep lakes with stable vertical stratification of the water mass.

The functioning of the ecosystems of large deep lakes is less dependent upon the landscape factor which is revealed mostly through the drainage. The relative value of river discharge (as the main criterion of the connection of a water-body with the drainage area) is expressed as a ratio of annual discharge (w-annual) to the volume of water in the lake (v) – $K = \frac{w}{v}$. In the small and large but shallow lakes of North-West Europe this value is usually greater than 1, and in some lakes it reaches 4 (Lake Ilmen) or even 7 (Lake Kubenskoe). In large deep lakes the annual water exchange is 10-100 times smaller. For instance, in Lakes Ladoga and Onega the annual water exchange equals 0.07; and in Lake Baikal it is only 0.003. Corresponding to this, the value of nutrient loading per unit surface area in a large deep lake (under the same landscape conditions) is many times less than is small lakes. This relationship is considered as one of the main ones when estimating nutrient loading upon a water body (Vollenweider 1975; Dilion & Rigler 1975).

The very low magnitude of the mobilisation of nutrients per unit area of a deepwater lake is known to be the main cause of oligotrophy in a water body. Another essential factor of oligotrophy and the associated with it high water quality in large deep lakes, is stratification of the water mass and low water temperature in the hypolimnion and bottom sediments, which slows down nutrient regeneration and migration of nutrients into the photic zone.

2. Water quality formation in every water body takes place under heterogenous ecological conditions. This is especially well manifested in large lakes with complex morphology. Let us consider the main manifestations of the ecosystem heterogeneity of large deep lakes which should be taken into account during ecological monitoring on the corresponding water bodies.

On the background of low total biological productivity in a large deep lake there are observed sharp differences in the productivity between separate regions and zones. The shallow zone, especially in the presence of some morphological isolation from the zone of deep water, is distinguished by higher productivity at all the trophic levels. This is due not only to accelerated nutrient regeneration here (because of the higher temperature of the water and bottom deposits) and their unimpeded mobilisation into the photic zone, but also due to more intensive mobilisation of the nutrients from external sources – from the drainage area and from the hypolimnion by upwelling. The latter has been well traced in Lakes Ontario and Michigan. During upwelling the near-bottom waters, entering the shallows from deep areas, contain increased concentrations of phosphorus, nitrogen and silicon compounds, which cause stimulation of photosynthesis in the shallows (Shelske et al 1971). However, in colder and more oligotrophic lakes (eg Lake Onega), where nutrient regeneration in the hypolimnion and the bottom is very slow, not an increase but a decrease in phytoplankton productivity in the corresponding zone is often observed during upwelling.

Shallow and deepwater zones of large lakes differ, not only in the level of biological productivity of communities (and consequently, in water quality) but also in the composition of populations and the structure of communities. Seasonal cycles of plankton are essentially different in them. In the annual phyto- and zooplankton cycle in the shallow zone there are, as a rule, two or three peaks in the biomass and, in a deepwater zone one, or rarely, two. For example, according to K. Damman (1966) the curve of the total abundance of the phytoplankton in the deep area of Lake Michigan over the long period of observations (1928-1967) had only one peak (in July-August), and in the shallow region, two peaks (in May and in October-November). A similar phenomenon in the seasonal cycle of the zooplankton has been noticed in Lake Onega.

In connection with an ecological heterogeneity of large lakes the production processes in each limnological zone are to a great degree determined by the peculiarities in the seasonal cycles of migration of substances within the water body. The quite different intensity of the processes in different seasons and years is apparently the main cause of the discrepancy between the great annual fluctuations in productivity of plankton communities in water bodies of this size and the comparative stability of their hydrological regimes. This paradox is especially notable in the Baikal where, for instance, the differences in the numbers of the main phytoplankton species produce a ratio of 200 to 1 between productive and unproductive years and that of the annual fluctuations of the zooplankton biomass reaches 10:1. (Antipova et al, 1970).

The deep part of a lake is also not uniform limnologically. The usual limnic zones: littoral, sublittoral and profundal are differentiated in it to some degree. In the annual limnological cycle of the pelagic zone there can be revealed the characteristic hydrological structures associated with formation of the thermal bar in spring and autumn, and the thermocline in summer. With the establishment of the thermal bar in spring, a thermally active, rapidly heated shallow zone and a thermally inert deepwater zone, maintaining the winter regime (having water temperature below 4°C), are formed in a lake.

The former gradually expands and the latter decreases and finally disappears by the beginning of summer; at this time it persists only as a hypolimnion (Figure 1). Such a hydrological structure influences greatly the course of biological processes determining the direction of population successions and the rhythms in the productivity of phyto- and zooplankton. It also influences the formation of water quality. The thermal bar restricts mixing of the lake water and all the water inputs (both river and domestic or industrial ones) at this time spread into the thermally active in-shore zone. Since, in the presence of the thermal bar, a rapid development of plankton takes place here, the quality of water in this zone decreases markedly.

In summer, in almost all deep lakes of temperate latitudes, stratification of the water into epilimnion, metalimnion and hypolimnion occurs. The water temperature in the epilimnion reaches 16-20°C, in some lakes — 22-25°C, and determines the high intensity of the biological processes and the greatest species diversity in the plankton. In some water bodies a water bloom caused by the blue-green algae is observed sometimes in the epilimnion, but, in the majority of deep lakes, the diatoms predominate in the phytoplankton even in summer. The hypolimnion, by contrast, has a low temperature during the whole summer season and is very poorly populated.

According to the indices of biological productivity of the pelagic and benthic communities, the hypolimnion in the lakes Ladoga, Onega and in many other deep lakes can be classified as ultraoligotrophic water. However, in the evaluation of water quality of this zone, one should take into account the possibility of accumulation of the diatoms (mainly *Melosira* species) which have sunk down after the spring outburst of phytoplankton. At the beginning of autumn, due to the vertical mixing of the lake water, these algae rise into the photic zone again and form a second, but less well-pronounced, vegetation outburst. This phenomenon has been observed in the Lakes Onega and Ladoga. (Petrova, 1968, 1971).

We have still to make clear how widely distributed the hypolimnic accumulations of the diatoms are and how they affect the lake water quality.

3. The enumerated parameters (far from all) of the heterogeneity of the ecosystem of large lakes allows us to recognise that water quality formation takes place under strongly varying temporal (seasonal phases) and spatial conditions. Therefore in accordance with final aim of ecological monitoring of aquatic ecosystems – the prognosis of the water quality, it is expedient to study not only factors and biological indices of variability of an ecosystem but also the laws of its resistance ie stability of an ecosystem in relation to changing external conditions. The first aspect (processes of variability) is seen better and more intensely studied in small water bodies; the second one is more strongly manifested in large-scale ecosystems, but the corresponding mechanisms are still little investigated.

The main mechanism of ecosystem stability, (most exactly, flexibility,) is generally accepted to be the ecological plasticity of species, which assures a relative stability in population composition and ecosystem functioning manifested in a certain type of circulation of substances. It is well known, however, that the ecological plasticity of each species is bound within its own limits in relation to certain factors (specific to a given species). These limits are very narrow. On the background of a very wide range of tolerated environmental variability, the ecological optimum at which a population is highly productive is, as a rule, confined to a very narrow range of vital factors. The limitation in the ecological plasticity of species is confirmed by a well developed system of population successions (in time and space) which are especially well marked in the communities of limnic plankton.

However, population change, in relation to the gradient of one or other factor, by itself does not assure ecosystem stability.

Ecosystem functional stability under changing external conditions is determined by the population changes (successions) being accompanied by their functional interchangeability within the limits of a given trophic level.

In the analyses of ecosystem stability (more correctly, flexibility) one may take into account the following;

1. Ecological specificity of a species;

2. Species diversity of an ecosystem, and;

3. *Unitary character of the structure of ecosystem functioning.*

In the circulation of matter, as in the basic process of ecosystem functioning, two compulsory (for each ecosystem) phases may be distinguished: autotrophic and heterotrophic. In principle, an ecosystem may function normally when only two populations are present – one autotrophic, which assures the synthesis of organic matter (for example one algal species), and one heterotrophic, which assures the destruction of organic matter (for example, one bacterial species).

But in the natural ecosystems there are always very many populations, both autotrophic and heterotrophic. This ensures wide possibilities for successions and functional interchangeability of populations which determine the main factor of an ecosystem's stability when the external conditions change. Besides, the species or, to be more correct, population-ecological diversity of the biota provides interconnection of the processes in different parts of a heterogenous water body as a *single ecological system.* The essential role in the mechanism of the functional interchangeability of populations is apparently played by competitive relations between them.

Each population in nature exists in the state of competitive stress with some other populations of a community. This fact provides the continuity of succession and its development even with very small changes in ecosystem conditions.

Competitive relations exist also between communities, as well as between taxocenes. A competitative relationship is well known to exist between the communities of micro- and macrovegetation (phytoplankton and macrophytes). Since the competition for nutrients between these communities effects the formation of water quality, let us dwell on this at greater length. T N Pokrovskaya (1974) has shown the following changes in the interrelationship between phytoplankton and macrophytes in the process of eutrophication of a shallow water body. The macrophytes are better able to compete for nutrients, and until the time when they lose the ability to increase their productivity at the expense of the external source of nutrients, they will hold back any noticeable changes in the productivity of phytoplankton. But as soon as the productive capabilities of the macrophytes are exhausted their protective function of pelagial eutrophication will stop. Part of the nutrients from external sources, on entering the trophic layer, will begin to be used by the phytoplankton. As a result, photosynthetic activity and phytoplankton productivity increase and cause a decrease in water transparency and a reduction of macrophyte productivity. In some cases it has been proved that a depression of the macrophytes (under the influence of high nutrient loading) occurs due to mass development and shading by the epiphytes and filamentous algae. This phenomenon has been well described also for fertilised ponds (Phillips, et al, 1978).

The ratio of productivity of macrophytes to phytoplankton in the total balance of photosynthesis of organic matter in a water-body influences significantly the ecological structure and type of circulation of matter in a limnic ecosystem. It has been shown by the example of the north western USSR water bodies (Nikolaev, 1977) that when the macrophytes are poorly developed (covering less than 15% of the surface area) the planktotrophic cycle predominates in the trophic system, but if the macrophytes cover more than 20% of the surface area, the benthotrophic system predominates (Figure 2). The ecosystems of all the deepwater lakes are planktotrophic ones. The same type of ecosystem may be found also in shallow water bodies, but the latter must be large in surface area and have a low water transparency as, for instance, Lake Beloe in the Vologodskaya region (mean depth 4 m transparency 1-1.5 m).

4. At the present time various methods of evaluation of water quality by biological indices have been proposed. They can be grouped into three main categories: evaluation by indicator organisms, by community structure and by functional indices of the activity of biological processes. Let us consider some of them.

The indicator organism method of evaluation of organic pollution of water or, as it is often called, saprobiological analysis of water proposed by Kolkwitz and Marsson (Kolkwitz & Marsson, 1908, 1909) at the beginning of our century, is the best known. This method has been perfected by several authors and now exists in various modifications (Knopp, 1955; Liebmann, 1962; Pantle & Buck, 1955; Sladecek, 1972; Woodiwiss 1964 and others).

In spite of the very detailed modern saprobiological analysis (for example, the list of saprobic species in Sladecek's paper contains about 2000 names) (Sladecek, 1973) and its seeming universality for application to all types of water bodies, this method is used in the practice of evaluation of water quality only in certain (mainly European) countries.

The main difficulty for the practical application of this method is an insufficient elaboration of the taxonomy of the aquatic fauna and flora. Identification of many species of plankton, benthos and other communities is impeded by the everlasting changes by systematists of their taxonomic status.

In taxonomic reference books the species become covered with a 'coat' of synonymy. The difficulties are aggravated by the fact that modern taxonomy is not adapted to ecological investigations. The descriptions of many species are given by features which need a special treatment of an organism for their recognition. During the ecological investigations, where prompt quantitative evaluation of a corresponding population is needed, exact determination of the species becomes extraordinarily labour-consuming and sometimes, when there are several morphologically similar species in a sample, a wholly impracticable operation.

In large heterolimnic water bodies only the populations of benthic and periphyton communities may be used for evaluation of water quality by indicator organisms. The use of plankton species for this purpose becomes very complicated due to intensive internal water exchange. The latter facilitates variously directed and often quite intensive migration of plankton organisms. As a result, populations can be found for a short time in water not at all corresponding to their ecology.

Among structural ecological indices of water quality, much attention has been paid in the literature to the index of species diversity and the role of separate taxonomic groups in communities (for instance Oligochaeta in zoobenthos). Both indices are unsuitable for large oligotrophic lakes. For example, by the index of species diversity (both in plankton and in benthos) the water of the hypolimnion in lakes Ladoga and Onega should be considered highly polluted; in fact it is very pure and may be used for drinking water supply almost without treatment or after slight treatment for seston (during spring and autumn mixing up).

The index of abundance of Oligochaeta in the benthos is also unsuitable for these purposes. In both the mentioned lakes, the species diversity and the role of this group in profundal benthos are very considerable (Gerd, 1950).

A change in species composition and taxonomic structure of communities, ie, in their temporal and spatial successions, is a more reliable biological index of disturbances in ecosystem.

In communities of a stable ecosystem (under stable climatic and external conditions) two types of succession are distinguished: 1) temporal (seasonal), when species composition changes during the annual cycle, and 2) spatial successions — local changes of species composition within the boundaries of a water body. In the anthropogenic disturbance of an ecosystem (for instance under the influence of increased nutrient loading or as a result of hydrotechnical changes in the aquatic regime) an especial type of succession appears — one over many years (long period).

The many-year successions under the influence of increasing nutrient loading, reflecting the course of anthropogenic eutrophication, are well studied and described for many of the world's lakes. This phenomenon has been followed in an especially detailed way in the plankton communities of large Swiss lakes.

When using this criterion in the practice of ecological monitoring one must take into account the following:

1. In the development of many-year succession of any community both common characters for different water bodies which are revealed in a certain succession of large taxonomic groups (eg in an increase in the role of the blue-green algae in phytoplankton with an increase in nutrient loading) and individual ones — succession of certain species peculiar for a given water body are distinguished.

2. For a reliable conclusion about many-year succession an unbroken series of observations for not less than 4-5 years is necessary in order to detect inter-year changes in taxonomic composition, which, in plankton communities, are observed even in relatively stable ecosystems of very different water bodies. For example, in the Baikal phytoplankton *Melosira* species prevail in

some years and *Cyclotella* and *Synedra* in others. Successions of species composition in the process of eutrophication and respective deterioration of the water quality appears especially clearly in the phytoplankton community. This is observed not only in small but also in the largest water bodies. In respect to the latter our data are in full agreement with Stoermer's opinion (1978) that in large heterogenous ecosystems such as the American Great Lakes (and Lake Ladoga and Onega in Europe — I N), the species composition of phytoplankton is an integral index of the water quality, the importance of which is still underestimated. The author's comment that application of this index for the evaluation of the trophic status of the Great Lakes is limited by the lack of taxonomic knowledge about them is true of the most other lakes in the world.

Judging by species composition of the phytoplankton and its seasonal succession, as well as by the biomass of this community, Stoermer (1978) established the following trophic gradations in the American Great Lakes: Lake Superior — oligotrophic, Lake Huron — mesotrophic, Lakes Erie and Ontario — eutrophic. Judging by the same indices and by chlorophyll content, Lake Onega is oligotrophic, and Lake Ladoga is slightly mesotrophic (Nikolaev and Petrova, 1978). Different parts of large lakes may, of course, have a very different trophic status and water quality. For example, the northern part of Lake Michigan is mesotrophic, the southern part slightly eutrophic, and some bays are eutrophic (Stoermer, 1978).

The species composition of zooplankton, which is more conservative, is less suitable for indicating changes in trophic condition of a water body. In Lake Ladoga, for example, the basic species composition of the zooplankton has not changed for the last 70 years (1906-1975-76), but notable changes have taken place for the same period of time in the dominant species of phytoplankton and new species have appeared. (Nikolaev and Petrova, 1978). However, if the process of eutrophication of water is rapid, a directed succession is well pronounced also, in the zooplankton. For example, great changes in the crustacean plankton composition have taken place in upper Bodensee in the 7 years, 1951-57 (Einsle, 1978).

For evaluation of the trophic level of water-bodies and for detection of eutrophication tendencies the indices of nutrient loading (primarily of phosphorus and nitrogen), as well as the indices of phytoplankton, photosynthesic productivity and (or) chlorophyll content are widely used at present (Vinberg, 1960) (Likens 1975) gives the most generalised gradation of the respective indices. (Table 1).

In publications on hydrobiology of lakes there is much specific information on productivity of other communities, including the abundance of bacterio — and zooplankton. This information allows to make an additional and not less reliable evaluation of the status of a water-body basin on trophic gradations (Tables 2, 3).

The indices given in tables 1-3 are lacking in precision. Most of all this concerns the time (month or season of the year) for which the figures concerned are given (tables 1, 2). In further investigations the corresponding indices of productivity of autotrophic and heterotrophic communities should be defined for specific limnic regions. It is very important that the concept of oligo- meso- and eutrophy of a water body for various geographical zones is associated with a different gradation in the intensity of bioproduction processes and corresponding indices.

Table 1. Generalised characteristics of lakes of various trophic status (according to Likens, 1975)

Trophic status of water body	Total P mg/m^3	Total N mg/m^3	Phytoplankton biomass $mg/C/m^3$	Primary production $mg\ C/m^2.day$	Chlorophyll mg/m^2	Dominating phytoplankton
Ultraoligotrophic	1–5	1–250	50	50	0.01–0.5	Chrysophyceae, Cryptophyceae
Oligotrophic	–	–	20–100	50–300	0.3–3	Dinophyceae, Bacillariophyceae
Oligo-mesotrophic	5–10	250–600	–	–	–	
Meso-trophic	–	–	100–300	250–1000	2–15	
Meso-eutrophic	10–30	500–1000	300	600–8000	10–500	Bacillariophyceae, Cyanophyta, Chlorophyta, Euglenophyta

Table 2. Quantitative indices of trophic level of the European USSR water bodies according to phytoplankton photosynthesis and total bacterioplankton density (according to V I Romanenko, 1965).

Type of water body	Photosynthesis $gC/m^2/day$	Total number $m10^6 n.cell/ml$	Water bodies
Oligotrophic	0.01–0.1	0.2–1	Lake Onega, Baikal
Mesotrophic	0.5–2	1–3	Rybinskoe, Gorkovskoe reservoirs
Eutrophic	2	4	Tsymlyanske, Kremenchugskoe reservoirs

Table 3. In a later work, V I Romanenko (1971) provides microbiological indices of water quality according to the ratio the number of saprophytic bacteria growing on MPA to the total number of bacteria in the water:

Number of saprophytes	Total number of bacteria	Water quality	Water body
0.003	or less	Very pure	Lake Onega, Baikal
0.03		Pure	Rybinskoe, Cherepovetskoe, Kuibyshev reservoirs
0.3		Polluted	Some parts of water-bodies
3	or more	Heavily polluted	Collectors of sewage and waste waters near towns.

5. For practical application of methods of indication of trophic status (and water quality) of a water body preference should be given to those integral indices which, still being sufficiently reliable, could be obtained with the least effort, ie using instruments with automatic registration of the needed information about a water body. In this respect the methods of evaluation of the activity of biological processes by oxygen seem promising. It was established long ago that the degree of oxygen saturation depends on two variables: intensity of gaseous exchange between water and atmosphere and intensity of biological processes; physical processes being of the most importance in oligotrophic water bodies, and biological ones in eutrophic water bodies. The intensity of biological processes may be determined by the rate of enrichment of the water with oxygen in the process of photosynthesis and (or) by the rate of biochemical consumption of oxygen. Both criteria are taken into consideration in the practice of evaluation of water quality and trophic status of a water body, but only in discrete determinations of the indices concerned.

The method of evaluation of an activity of biological processes (and water quality) by the indices of seasonal changes in the oxygen balance in a water body, which is more integral and expedient in our opinion for the practice of ecological monitoring, has not been properly applied. The index of the rate of the consumption of oxygen in biochemical processes during winter (from the autumnal overturn, when the maximum saturation of water with oxygen is observed to the end of winter, when there is the minimum oxygen content in it) deserves special attention.

Hutchinson (1957), in his thorough survey of the oxygen regime in lakes, cites highly expressive indices of the areal (ie with regard to the morphometry of a water body) oxygen defict in the water for two groups of lakes: 50-75 metres and 20-25 metres deep.

These indices are very generalised for evaluation of the trophic level of a water body and water quality in it, but they deserve attention for the purpose of monitoring because the information concerned (about oxygen content in the water) may be obtained by means of automatic instrumentation.

Another criterion — oxygen hysteresis — may be also used for deep oligotropic lakes. This has been shown by Forsh — Menshutkina (1973) in the case of Lake Onega. The annual course of dissolved oxygen content under the oligotropic conditions in Lake Onega is determined not by the process of phytoplankton photosynthesis and what is more — not by processes of the breakdown of organic matter but by a physical process — namely by the intensity of gaseous exchange between water and atmosphere. The course of this process is influenced by water temperature and by the effect of the delay in the exchange of oxygen between atmosphere and water in comparison with changes in oxygen saturation concentration. The consequence of this delay is supersaturation of water with oxygen during the phase of its warming and oxygen deficit during the phase of cooling, (the phenomenon of oxygen hysteresis). Peculiarities of this process are presented by the above-mentioned author in the form of a phase diagram with the following comments (figure 3). The basis of the idea of oxygen hysteresis is an assertion of the proportionality of the influx, or efflux, of a dissolved gas to the difference in partial pressures of this gas in the liquid and in the atmosphere. In connection with the fact that the function of the dependence of oxygen saturation on water temperature is a simple one, the difference in partial pressures is presented on the graph shown as a variable difference between concentrations: $C_t - C$, where C_t — concentration of dissolved oxygen in water at the moment of observation, and C — oxygen concentration, corresponding to saturation at the observed temperature. On the graph the abscissa axis is for the C_t value, the value of C and its corresponding temperature are plotted on the ordinate axis. It is taken into account that the relationship between temperature and oxygen concentration at saturation is nonlinear. On the phase diagram the bisector of the coordinate angle corresponds to the condition of saturation of the water with oxygen ie, $C = C_t$. The points which correspond to the phase of warming of water (June, July, August) are situated above the bisector of the coordinate angle, which corresponds to the condition of supersaturation of water with oxygen; and the points which correspond to the phase of cooling of the water are situated below the mentioned line, which indicates oxygen deficiency.

It is worth mentioning that in this case the deficit of oxygen differs essentially from the oxygen deficit related to breakdown of organic matter (observed in eutrophic waters). The difference is that, in the first case (in the case of hysteresis), the oxygen deficit is observed when the absolute oxygen content in the water increases, but in the second case it actually decreases.

For oligotrophic (lakes Onega and Teletskoe) water bodies, the oxygen curve is harmonic (Figures 3, 4) and for eutrophic ones (Lake Dalnee) it is considerably distorted (figure 5).

From the point of view of ecological monitoring the deviation of the seasonal cycle curve of oxygen hystereses from the normal one for an oligotrophic lake may be used as an index of changes in the lake towards eutrophy, since the deformation of the oxygen curve is determined by an increased activity in the biological processes ie, it reflects a tendency towards eutrophy.

The possible determination of seston density with the help of technical aids also deserves attention for application in the practice of ecological monitoring. On large lakes with a heterogenous ecosystem it is convenient to contour zones with increased seston density, which is represented mainly by plankton organisms in the majority of the water-bodies, with the help of instruments (installed on board a ship).

The perfection of the instruments for quantitative determination of the chlorophyll content of phytoplankton also opens up great prospects for these same purposes.

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Figure 1. Seasonal phases of limnological structure of Lake Onega.

I – warmwater high-productive zones

II – coldwater low-productive zones

III – thermal bar

IV – thermocline

1 – spring; 2 – summer; 3 – autumn; 4 – winter

Figure 2. General trophic structure of a lake ecosystem.

1 — trophic relationship

2 — migration relationship

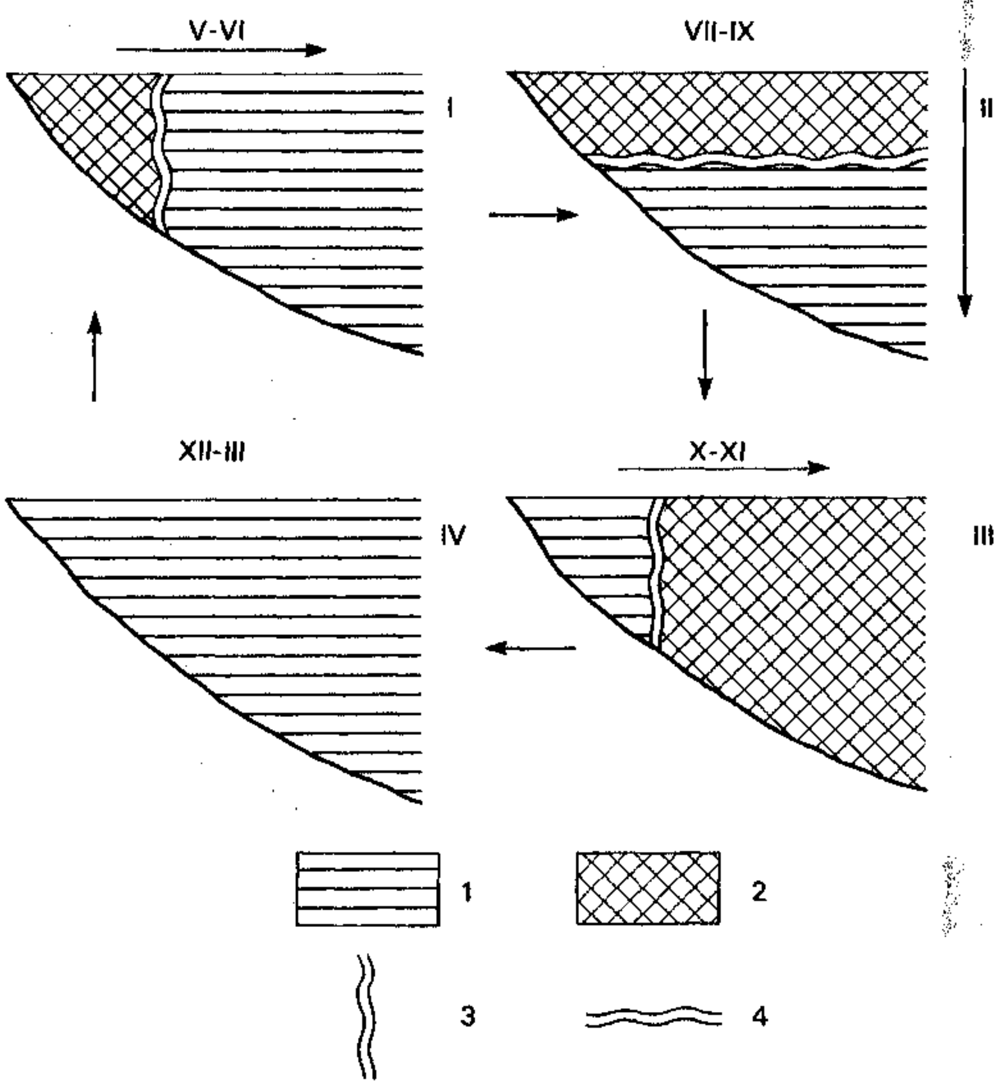

FIG 1

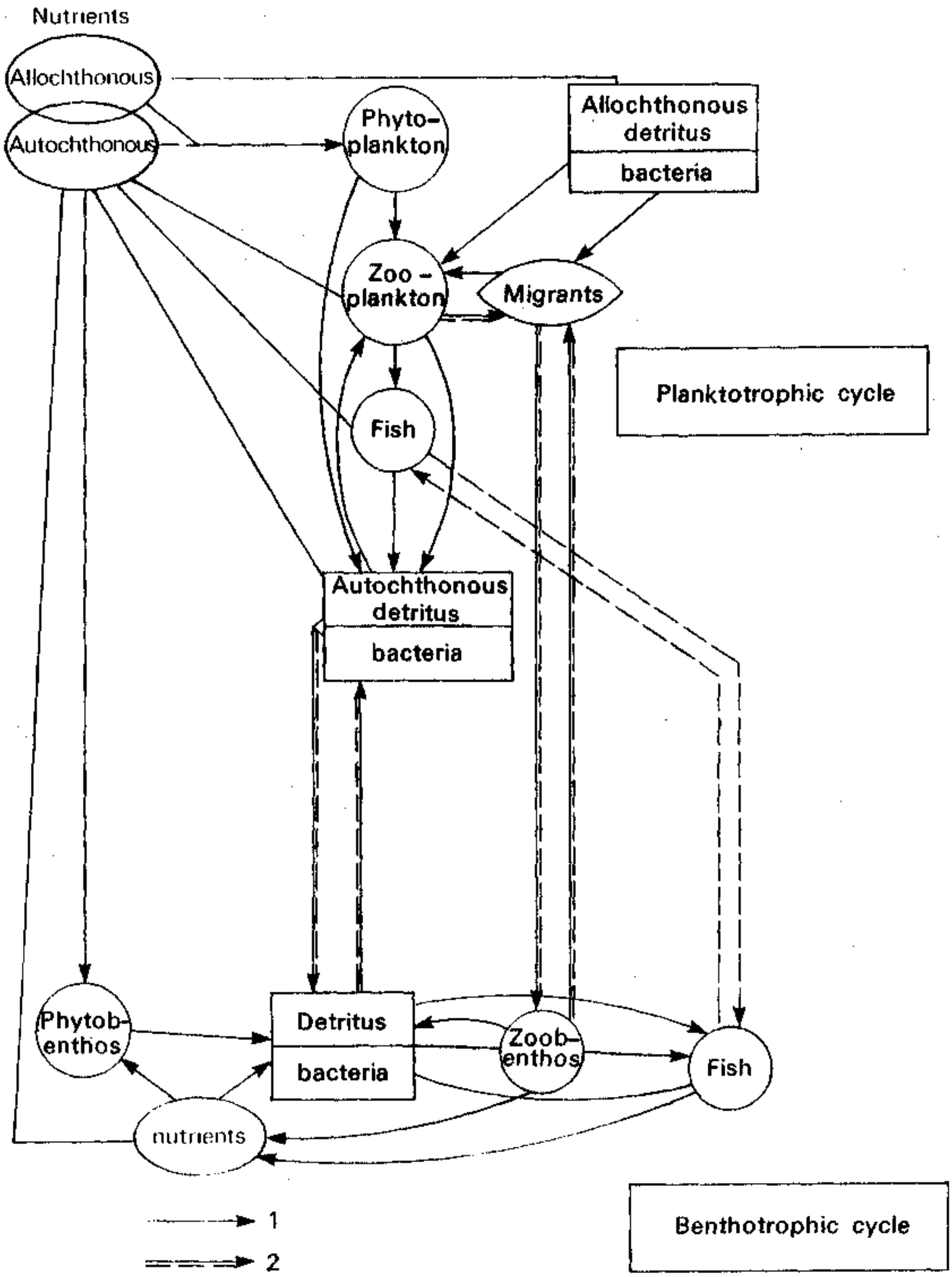
Nutrients
Allochthonous
Autochthonous
Phyto-plankton
Allochthonous detritus
bacteria
Zoo-plankton
Migrants
Fish
Planktotrophic cycle
Autochthonous detritus
bacteria
Phytob-enthos
Detritus
bacteria
Zoob-enthos
Fish
nutrients
1
2
Benthotrophic cycle

FIG 2

Figure 3. Curves of oxygen histeresis for surface-water of the central part of Lake Onega.

I – station 26: 1 – 10/VI, 2 – 26/VI, 3 – 17/VII, 4 – 20/VIII, 5 – 30/IX, 6 – 29/X, 7 – 2/XII – 1966.

II – station 4 (shallow water): 1 – 6/VI, 2 – 24/VI, 3 – 15/VII, 4 – 19/VIII, 5 – no observations, 6 – 29/X – 1966.

C_t – observed oxygen concentration;

C – concentration of oxygen corresponding to the saturation at the observed temperature

(according to T Forsh-Menshutkina, 1973)

Figure 4. Curve of oxygen histeresis for Lake Teletskoe.

1 – 2/VII, 2 – 18/IX, 3 – 2/XII – 1931;

4 – 31/VII – 1930; 5 – 11/X – 1932;

C_t – observed oxygen concentration

C – concentration of oxygen corresponding to the saturation at the observed temperature.

(according to T Forsh-Menshutkina, 1973).

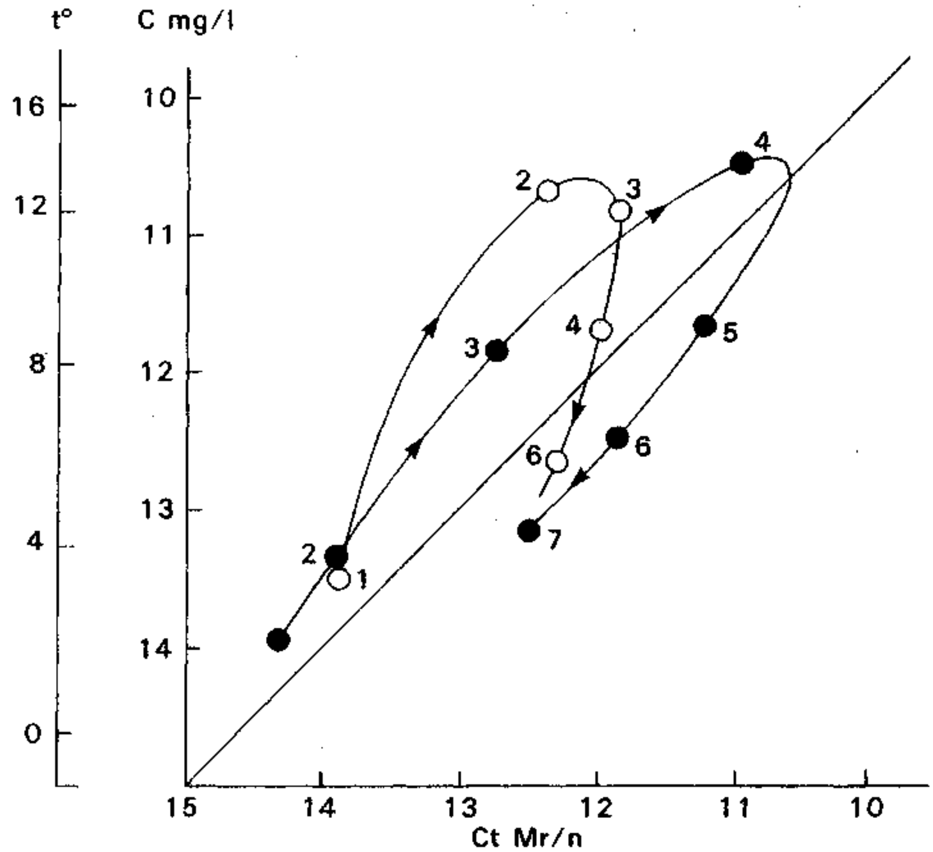
t°
C mg/l
16
12
8
4
0
10
11
12
13
14
15
14
13
12
11
10
Ct Mr/n
1
2
3
4
5
6
7

FIG 3

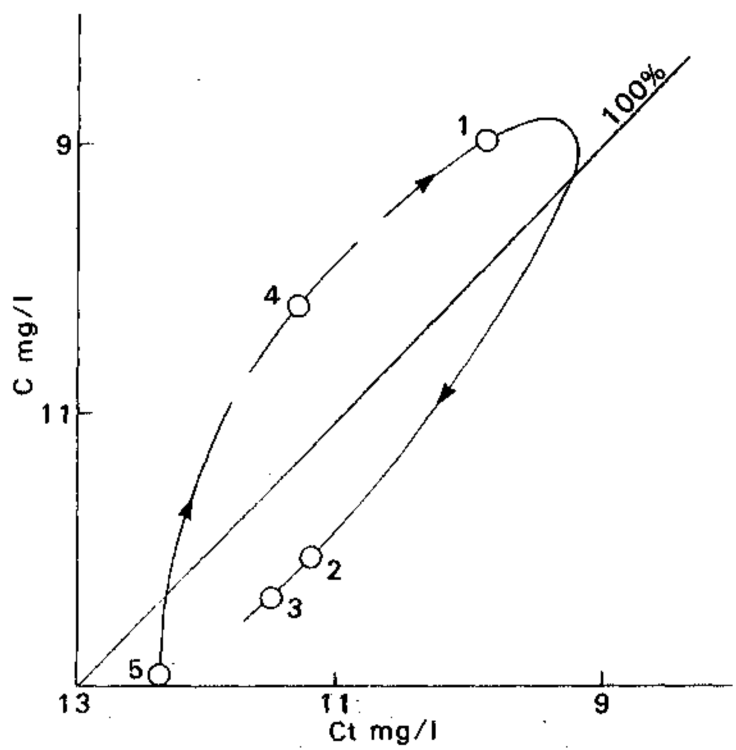
C mg/l
9
11
13
11
9
Ct mg/l
100%
1
2
3
4
5

FIG 4

Figure 5. Curve of oxygen histeresis for Lake Dalnee.

1 – 13/V, 2 – 3/V, 3 – 15/VI, 4 – 30/VI, 5 – 14/VII,

6 – 16/VIII, 7 – 30/VIII, 8 – 13/IX, 9 – 30/IX, 10 – 18/X,

11 – 10/XI, 12 – 9/XII – 1965.

C_t – observed oxygen concentration

C – concentration of oxygen corresponding to the saturation at the observed temperature.

(according to T Forsh-Menshutkina, 1973).

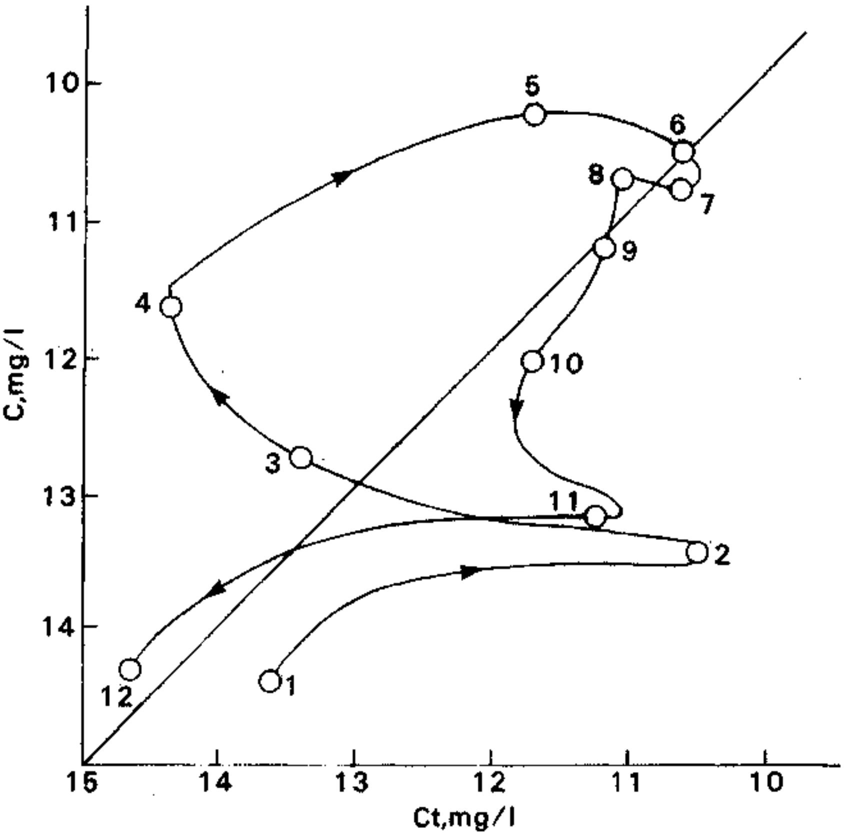

FIG.5